

Removal of Salt-killed Vegetation during Tidal Restoration of a New England Salt Marsh: Effects on Wrack Movement and the Establishment of Native Halophytes

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ABSTRACT

A New England salt marsh undergoing tidal restoration was manipulated to improve halophyte seed dispersal and encourage the expansion of salt marsh plant communities. I created ten openings (150 m²) in an area of dead freshwater shrubs and common reed (*Phragmites australis*), which had been killed by saltwater inundation during tidal restoration. The dead plants presented a physical barrier to the upstream movement of waterborne seeds from halophyte species. Five of the openings were extended to the edge of the adjacent recovering salt marsh, providing a clear passageway into the plots cleared of barrier vegetation. Another five remained as isolated clearings and five uncut plots served as controls. The establishment of salt marsh plants was greatly enhanced by removing the barrier vegetation. While plots directly connected to the salt marsh yielded the highest numbers of new halophytes, isolated clearings also had a beneficial effect. These responses suggest that barrier vegetation removal can facilitate seed dispersal, colonization, and succession in a salt marsh habitat, and provides an effective alternative to standard restorative approaches such as artificial seeding and planting.

Keywords: Cape Cod, common reed (*Phragmites australis*), cordgrass (*Spartina alterniflora*), Massachusetts, salt marsh, tidal restoration, wrack

Introduction

Salt marshes along the coast of New England have long suffered from various hydrologic alterations, one of which is the construction of dikes for mosquito control and land reclamation (Roman et al. 1984). The reduction of seawater flow through dikes triggers dramatic physical and biogeochemical changes upstream (Portnoy 1999). These inevitably include numerous forms of biological degradation, including the replacement of native halophyte communities (salt-tolerant plants typical of normally functioning salt marshes) by freshwater, brackish, and even upland

taxa, some of which are invasive and exotic (Warren et al. 2002). Many thousands of acres of salt marsh plant communities, which provide the basic habitat structure and food energy for associated estuarine and marine biota, have been lost in this way.

Re-establishing tidal flow into diked systems is becoming increasingly commonplace as a way to ameliorate some of this damage. In 1999, tidal restoration began on the Hatches Harbor marsh—an 80-hectare system in the Cape Cod National Seashore diked in 1930. The marsh has experienced increased seawater exchange through a series of culverts built into the dike in 1998. The culverts were fitted with adjustable doors that were opened incrementally over a period of six years. Maximum flow capacity was reached in 2005 through an opening

that is now 27 times larger (7.79 m²) than the original drainage pipe (0.29 m²). As a result, the tidal range in the restricted marsh has increased by 59%, with high tides approximately 30 cm higher than before the opening.

During the restoration, the vegetation in Hatches Harbor has responded favorably to the changing physicochemical conditions (Gwilliam et al. 2007). However, continued expansion of native salt marsh species may now be limited by physical processes related to seed dispersal. Specifically, the upslope movement of wrack by flood tides during the fall through spring is impeded by the presence of dead non-halophytic species that could not tolerate the increased salinity levels. Salt marsh wrack is composed of dead and detached aboveground plant biomass—mostly cordgrass (*Spartina*

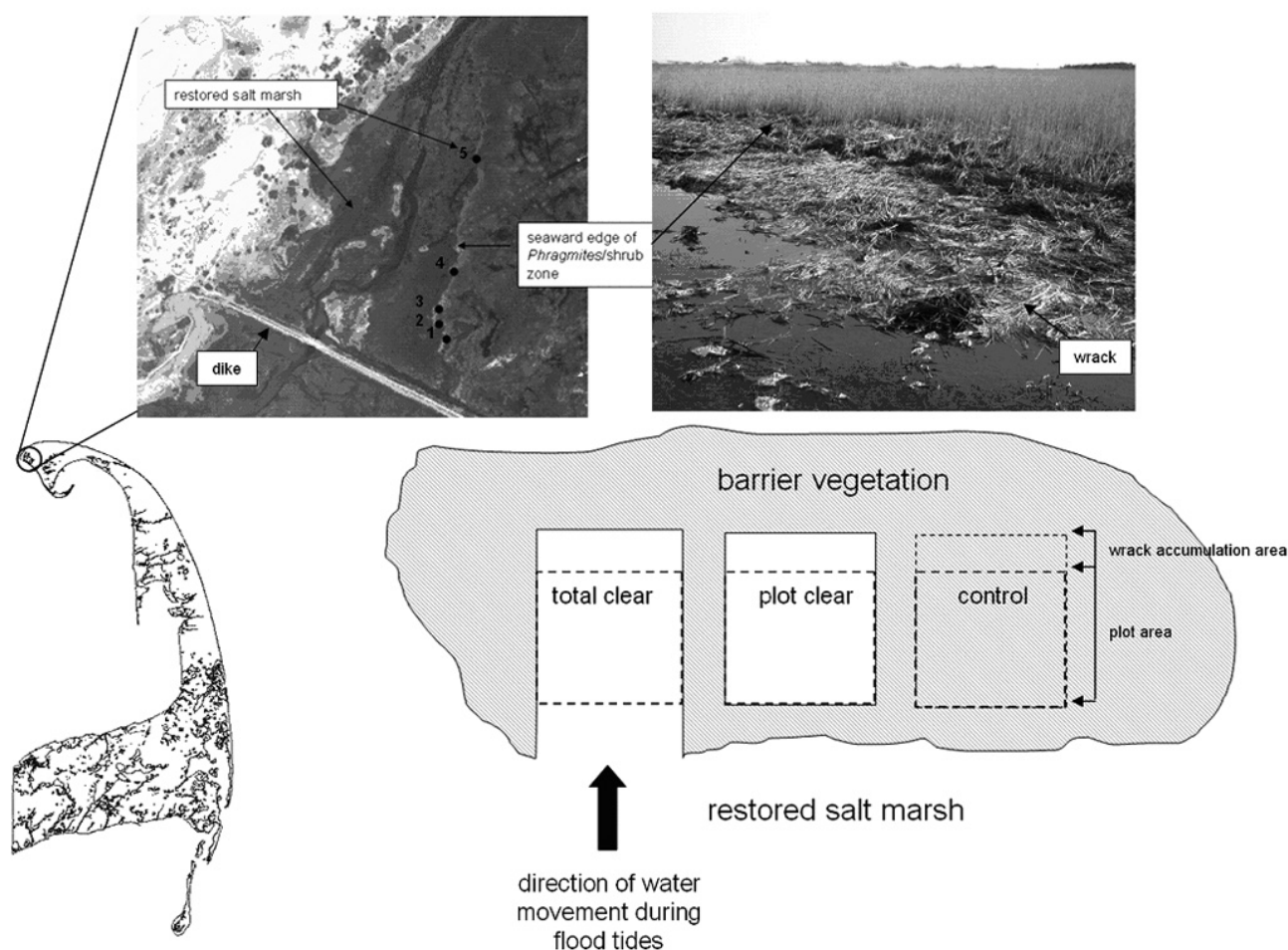


Figure 1. Map of Cape Cod showing location of Hatches Harbor. Blow up aerial of marsh, upper left, indicates locations of treatment sites (black circles). Bottom right drawing shows schematic of treatment design. The photo in the upper right corner shows typical accumulation of wrack at the boundary between the recovering salt marsh (foreground) and barrier vegetation (background); the latter consists largely of standing dead reed (*Phragmites australis*).

alterniflora)—from the previous growing season. This material tends to coalesce to form cohesive floating mats that trap a large number of seeds. The transport of salt marsh wrack across the tide-restored portion of Hatches Harbor is blocked by salt-killed common reed (*Phragmites australis*, an exotic grass), dead woody shrubs, and a lesser amount of live common reed—all of which had invaded the floodplain subsequent to diking of the marsh. The wrack accumulates along the front edge of this barrier vegetation and sometimes becomes entangled just a few meters into it (Figure 1). Thus, a large quantity of halophyte seed never reaches the interior marsh, even though physicochemical conditions are suitable for germination and

establishment. Furthermore, the barrier vegetation tends to keep old detritus (mostly dead stems of common reed) in place, rather than allowing it to be pushed upslope by the tides. A thick layer of thatch develops that limits the penetration of sunlight to the ground surface and limits seed germination.

Studies of seed dispersal in tidal wetlands are relatively scarce. Rand (2000) reported that seed distributions in a New England salt marsh suggest highly localized dispersal with very limited transport out of parental environments and that low seed availability influences plant abundance within marsh zones. Morzaria-Luna and Zedler (2007) similarly suggest that where seeds are limited, active

seeding and planting may be needed to encourage marsh development at restoration sites. Anecdotal evidence from the Hatches Harbor restoration project suggests that pathways through barrier vegetation encourage salt marsh plants to expand into the interior marsh. In 2004, two narrow tidal creeks were constructed in an attempt to recreate historic tidal creeks in the system. Shortly after these creeks were excavated, numerous salt marsh plants became established along the creek banks, presumably because the channels provided a conduit for seed transport through an otherwise significant physical barrier to wrack movement.

Methods

To assess the degree to which salt marsh recovery can be aided by the removal of barrier vegetation, I conducted a field experiment in the tide-restricted portion of Hatches Harbor. In this study, barrier vegetation was managed in different ways to facilitate wrack and detritus transport by tides and, ultimately, germination and establishment of native halophytes. In November 2005, five sites along the edge of the barrier vegetation zone in the tide-restored portion of Hatches Harbor (the seaward, salt-killed edge of which is conspicuous in aerial photography) were randomly selected using ArcGIS (ESRI 2003). At each site, the corners of three $10 \times 15 \text{ m}^2$ plots, spaced 5 m apart, were demarcated with PVC pipe. The area over which vegetation changes were to be evaluated was $10 \times 10 \text{ m}^2$. The additional $5 \times 10 \text{ m}^2$ area at the upstream end of the plot was created as a place for wrack and detritus to accumulate in order to minimize coverage within the plot. All plots were situated approximately 3–4 m from where the barrier vegetation abruptly transitions into recovering salt marsh (Figure 1). Interstitial salinities at these sites range between 28 and 33 parts per thousand (Gwilliam et al. 2007).

Before any treatments were applied, I assessed the cover of live and dead common reed and dead woody shrubs based on the Braun-Blanquet scale (0 = 0, >0–1% = 1, >1–5% = 2, 6–25% = 3, 26–50% = 4, 51–75% = 5, 76–100% = 6). Because of their low densities, individual plants of all native halophytes were counted, with the exception of saltmarsh hay (*Spartina patens*), where cover was estimated owing to the difficulty of counting the thin shoots that grow in high densities within a very small area. The quantity of wrack material that had already accumulated along the front edge of the barrier vegetation was also scored to obtain a surrogate measure of the availability of halophyte seeds that are associated with the wrack

masses. This was done by evaluating the entire wrack line at each site based on its average width and recording the amount: 0 = no wrack, 1 = <1 m wide, 2 = 1–2 m wide, 3 = >2 m wide.

After characterizing the initial plant communities, I applied the treatments. At each site, the vegetation in two plots (randomly selected) was cut with a sickle to near ground level ($\leq 15 \text{ cm}$) and left in place. Barrier vegetation between one of the plots and the adjacent salt marsh was also cut (total cut; TC) so that wrack could move freely upslope. In the other cut plot, the 3–4 m of barrier vegetation between the salt marsh and the plot was left intact (plot cut; PC). The third plot remained uncut, serving as the control (Figure 1).

In August 2006, the end of the growing season in southern New England, I reassessed the vegetation in each plot. At this time, the cover of any wrack or detritus that had not been transported off the ground surface of the plots was estimated in the same manner as vegetation, since this material reduces the amount of plot area over which seeds can germinate and survive. In fact, wrack accumulation can even be deleterious to mature stands of halophytes (Tolley and Christian 1999).

Spearman's rank order correlation was used to examine associations between non-parametric variables such as plant or wrack cover scores, using XLStat (Sokal and Rohlf 1995, Addinsoft 2005). Differences in the numbers of newly established halophytes among treatment groups were analyzed by one-way ANOVA after the data were square-root transformed to achieve normality and heteroscedasticity. Specific pairwise comparisons were then made using Tukey's HSD tests (StatSoft 2001).

Results

Composition of the original plant communities

Dead reed constituted the majority of barrier vegetation, with Braun-Blanquet cover scores averaging 5.2 and ranging between 3 and 6. Live reed accounted for much less cover and was less variable with all scores falling into the 2 or 3 categories. The cover of dead woody shrubs was the lowest and most variable among plots. The mean cover score for this component was 2.1, ranging between 0 and 4.

Prior to treatment, the abundance of all halophyte species was very low and statistically equal among groups.

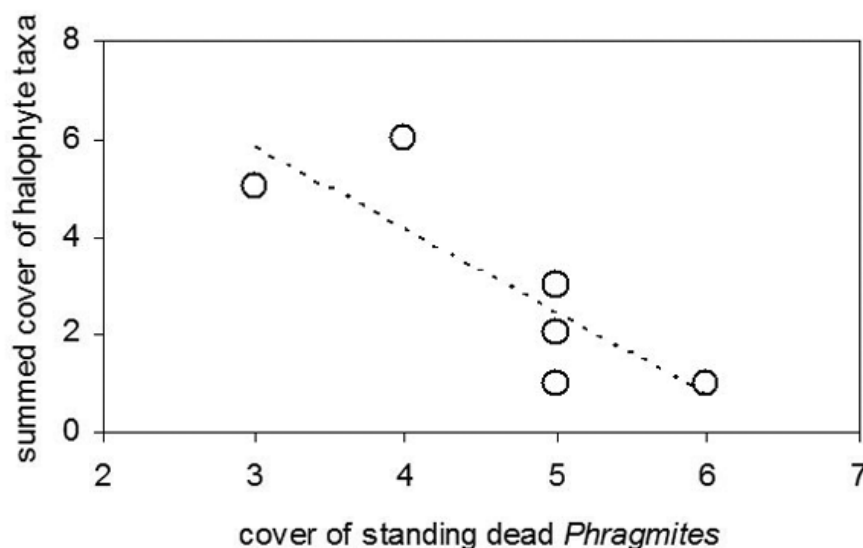


Figure 2. Relationship between pre-treatment cover of halophytes and the cover of standing dead reed (*Phragmites australis*).

Table 1. Mean (\pm SE) and total numbers of seedlings for each halophyte taxon recorded in September 2006. Cover class values are listed for salt-marsh hay; treatment groups with the same letter are statistically equal). Treatment effects were significant for cordgrass (ANOVA; $F_{2,9} = 13.1$, $p = 0.002$) and glasswort (ANOVA; $F_{2,9} = 5.31$, $p = 0.03$).

	Species	Pre-treatment			TC	Post-treatment			
		TC	PC	C		PC	C		
MEANS	<i>Spartina alterniflora</i>	0	0	0	468 (114)	a 172 (54)	a 26 (16)	b	
	<i>Salicornia maritima</i>	0	0.2	4.4	290 (109)	a 228 (106)	a 24 (15)	b	
	<i>Salicornia virginica</i>	0	0.2	0	0	a 0.2 (0.2)	a 0	a	
	<i>Suaeda maritima</i>	0	0.4	0.4	0	a 0.4 (0.4)	a 0.4 (0.4)	a	
	<i>Limonium carolinianum</i>	0	0	0	0	a 1.0 (0.63)	a 0.6 (0.6)	a	
	<i>Spartina patens</i> (cover)	0.4	0.6	0.6	0.6	a 0.2	a 0.6	a	
TOTALS	<i>Spartina alterniflora</i>	0	0	0	1873	860	128		
	<i>Salicornia maritima</i>	0	1	22	1158	1140	122		
	<i>Salicornia virginica</i>	0	0	0	0	1	0		
	<i>Suaeda maritima</i>	0	0	11	0	2	2		
	<i>Limonium carolinianum</i>	0	0	0	0	5	3		
	<i>Spartina patens</i> (sum cover)	3	3	2	3	3	1		

Nine out of the 15 total plots had no halophytes at all. With respect to individual taxa, smooth cordgrass (*Spartina alterniflora*)—the dominant species within the unrestricted and recovering salt marsh—was absent from all plots. Glasswort (*Salicornia maritima*) and annual sea blight (*Suaeda maritima*) were present, but relatively scarce. The former ranged between 0 and 11 individuals per plot, the latter between 0 and 10. In the 6 plots where halophytes were present, the summed cover scores of all taxa decreased with increasing cover of standing dead reed ($p = 0.02$), which comprised the bulk of the barrier vegetation (Figure 2). No significant associations with live reed or dead woody shrubs were found.

Wrack distribution

The treatments clearly affected the transport of wrack and detritus across the plots. In the TC plots, most of this material was carried across the plots by high flood tides and piled up along the upstream boundary. The same was true for PC plots although in these treatments the material consisted primarily of detritus from the barrier vegetation (mostly common reed stems) that had, prior to cutting, held it in place. In both treatments, however, variable amounts of wrack and detritus remained on the ground

within the plot boundaries. While a small amount of wrack and detritus appeared to have moved through some of the control plots, the vast majority of it remained piled up against the uncut seaward edge of the barrier vegetation.

Halophyte establishment

After numerous trips to the field sites over the course of the experiment, I noticed that one plot at the northernmost site (site 5) was a distinct outlier in terms of hydrologic conditions. Whereas all other plots were well-drained with exposed soil at low tide, this particular plot frequently remained flooded with approximately 10–15 cm of standing water owing to its lower elevation or topographic features that limit drainage. Presumably, this is why halophyte establishment was so low there. No other species except cordgrass became established in this plot, and only 10 individual seedlings were counted—a number that is more than 40-fold lower than the mean value for all other plots in that treatment group (TC). I excluded this plot from the analysis so that interpretation of the data would not be unduly confounded by this hydrologic anomaly. To compensate for this, a modified Tukey's HSD test for unequal N was used to test for post hoc differences among groups.

Comparisons of treatment means showed that the number of cordgrass seedlings present in both the TC and PC plots was statistically higher than in the control plots (Table 1). In fact, seedling numbers were roughly 18 and 7 times higher, respectively, than in the control group. In addition, there were more than twice as many seedlings in the TC compared to the PC group. A somewhat different response was observed for glasswort, although statistically the results were the same. Both the TC and PC treatments yielded very high and similar numbers of seedlings. Mean values for these groups were roughly ten times that of the control.

In addition to cordgrass, two new halophyte species appeared in the plots (Table 1). These were sea lavender (*Limonium carolinianum*) and perennial glasswort (*Salicornia virginica*). However, their numbers were very low and similar among treatments. With the exception of perennial glasswort and annual sea blight, the abundance of all species increased in the control plots, albeit to a much lesser extent than in the TC and PC treatments. The cover of saltmarsh hay remained essentially unchanged over the duration of the study.

When numbers of cordgrass or glasswort seedlings in each treatment group were plotted against the post-

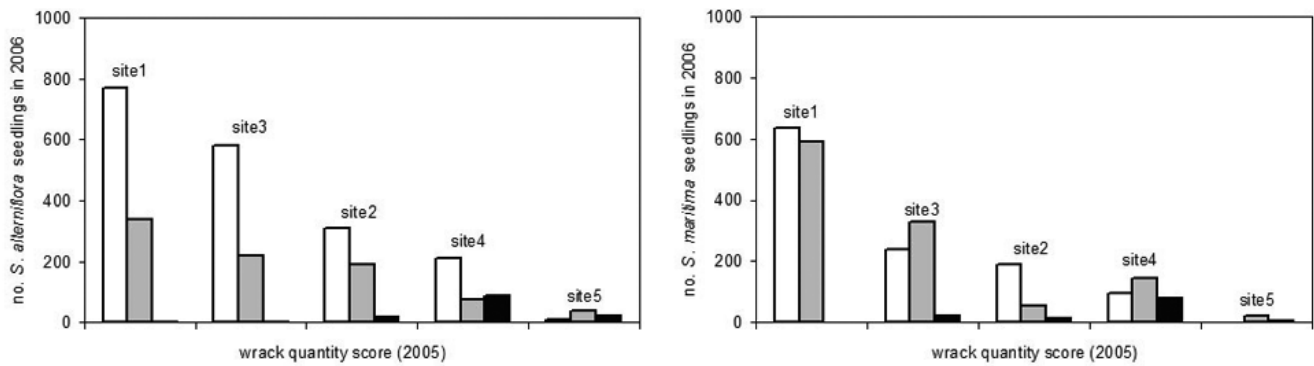


Figure 3. Number of cordgrass (*Spartina alterniflora*) seedlings in relation to pre-treatment wrack quantities by site and treatment (open bars = TC, shaded bars = PC, solid bars = controls).

treatment cover of wrack and detritus remaining within the plot area, no significant correlations were found. For both species, however, there were positive correlations ($p \leq 0.014$) between the number of seedlings in the TC and PC groups with the amounts of pre-treatment quantities of wrack at each site (Figure 3). In other words, the more wrack material that was potentially “available” to move across the plot, the higher the number of seedlings that eventually became established there.

Discussion

The pre-treatment abundance of native halophytes within the barrier vegetation zone provided preliminary evidence that standing dead vegetation was affecting salt marsh expansion across the tide-restored floodplain. Although the correlation was based on a small number of data points, numbers of halophytes were generally higher where the cover of barrier vegetation was lower. Experimental removal of barrier vegetation greatly amplified this trend. This was most evident for cordgrass, a key species of salt marshes along the entire Atlantic coast. The TC treatment provided a direct, unimpeded route for the transport of wrack into the zone of barrier vegetation and resulted in the largest number of cordgrass seedlings becoming established—more than double that of the PC group and over an order of magnitude higher than the control.

The PC treatment also encouraged establishment of cordgrass as well as glasswort at a level that was significantly higher than the control group. The mechanism for this is unclear for cordgrass given that this species has relatively short-lived seeds (less than 8 months) (Daehler 2000). As such, it seems unlikely that resident seed from previous years was able to germinate in the wake of detritus (mostly dead stems of common reed) movement out of the plot. An alternative explanation is that plot clearing provided both open passage and suitable germination sites (owing to the transport of reed detritus off the ground surface) for an unquantifiable number of free-floating seeds (seeds not associated with wrack deposits) that made it through the 3–4 m of barrier vegetation separating the area of recovering salt marsh from the plots.

Halophyte abundance in the control treatments also increased from 2005 to 2006. This apparently represents the rate of natural recruitment—either from seeds already resident in the soil that were able to germinate through natural openings in the detritus layer or from new, free-floating seeds able to penetrate the barrier vegetation. There was also a higher amount of rainfall in 2006 than in 2005, which can lower salinities and encourage halophyte germination (Shumway and Bertness 1992, Ungar 1995). Regardless, increases in the controls were minor (over 30 plants) compared to those in the TC and PC groups (nearly 150 plants).

Other extraneous factors undoubtedly contributed to variability in germination and establishment in this study. Flooding depth and duration are important since standing water can suppress germination (Broome et al. 1973, Ungar 1995). As Figure 3 demonstrates, other potential influences include considerable spatial heterogeneity in the amount of wrack passing through each plot. Moreover, certain edaphic conditions may favor or suppress germination (e.g., sediment properties, nutrient concentrations). Finally, some natural degradation of standing dead reed occurred during the study that also appears to have contributed to variability in the data.

Management Implications

This study indicates that salt marsh restoration may be steered by active vegetation management. In particular, the removal of barrier vegetation in the floodplain can greatly accelerate the spread of certain desirable species. Mowing or prescribed fire could also be used to accomplish this task on a larger scale, although in this study I was able to clear 1,500 m² (0.15 ha) with a sickle in about six to seven hours. Presumably, a large group of people with hand tools or a single mowing machine could clear large areas in a relatively short period of time. Under natural conditions, barrier vegetation may take many years to degrade to the point where adequate passage of wrack is possible. In fact, the dead woody shrubs that were cut down

in this study had been killed 6 years prior, after the initial reintroduction of seawater in 1999. Common reed deteriorates more quickly, but short stems that are high enough to inhibit or slow the passage of wrack often persist for long periods of time.

Clearing impediments to wrack movement is one means of promoting salt marsh colonization and establishment through natural dispersal and succession, and may be a good alternative to assembly approaches, such as planting and artificial seedling. In selecting appropriate locations for this type of management, however, marsh geomorphology, patterns of wrack movement, timing, and hydrology should be considered. For example, removing barrier vegetation in an area that tends to accumulate large amounts of wrack and that does not retain standing water for long periods of time would provide the best chance for success. Furthermore, any removal of barrier vegetation should be done shortly after the period of seed release (August to late September in southern New England). That way, fresh seed can move freely upstream immediately after it is dropped, rather than accumulating in immobile wrack lines where desiccation can substantially reduce viability (Broome and Seneca 1974). An ancillary benefit of removing barrier vegetation is an improvement in the aesthetic qualities of the landscape. This may be particularly important in systems where the tidally-restricted floodplain has been heavily invaded by woody vegetation (Portnoy and Reynolds 1997). Large areas of dead

shrubs and trees that have succumbed to elevated salinities and flooding can be disconcerting to some. Thus, actions that facilitate seed dispersal may also improve public perception and foster a wider acceptance of tidal restoration projects.

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